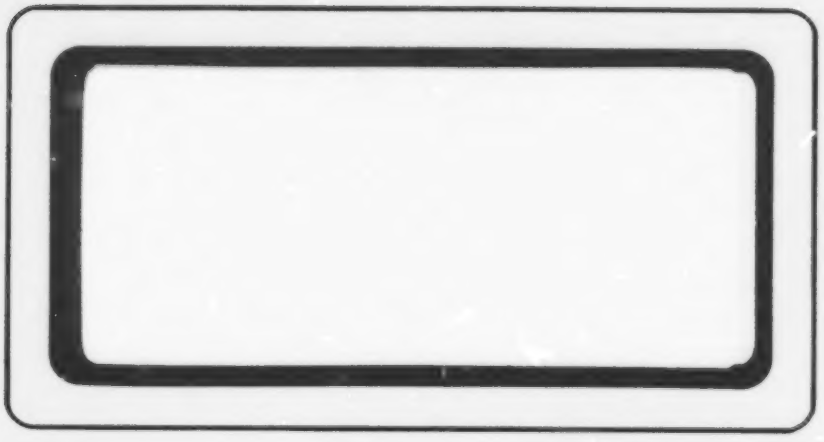

Province of British Columbia

Ministry of Environment and Parks

The Effects of Decreasing Heavy Metal Concentrations on the Biota of Buttle Lake; Vancouver Island, British Columbia

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Abstract

Since 1966, a copper-lead-zinc mine operated by Westmin Resources Limited has been a source of heavy metal input into Buttle Lake. Metal levels increased in the lake, peaking in 1980/81; thereafter improved treatment and collection systems at the mine site have resulted in steadily decreasing metal levels throughout the lake system. The increase in metal concentrations in the lake water was accompanied by increased metal concentrations in salmonid muscle and liver tissues, by elevated levels of hepatic metallothionein, and by declines in both species diversity and population for phytoplankton, periphyton and zooplankton. However, recovery of the biota of Buttle Lake with decrease in metal concentrations has not followed a simple reversal of the earlier trends.

While metal levels in rainbow trout muscle tissue have improved significantly, copper and cadmium liver tissue levels remain significantly higher than for the control lakes. Hepatic metallothionein levels in rainbow trout have declined steadily from a maximum of 269 $\pm$ 23 nmoles/g in 1981 to a low of 64 $\pm$ 22 nmoles/g in 1985, which is similar to the control lake.

Interspecies comparison of muscle and liver tissue levels both for Buttle Lake and for uncontaminated British Columbia lakes found that rainbow trout (Salmo gairdneri), cutthroat trout (Salmo clarkii clarkii) and Dolly Varden char (Salvelinus malma) respond differently to metals in the environment. It is apparent that trends in rainbow trout data should not be generalized to include all three groups.

As metal concentrations decreased, the phytoplankton community began to change so that beginning in 1983 and continuing well into 1985, a continuous phytoplankton bloom consisting of a virtual monoculture of Rhizosolenia eriensis was present throughout the lake. Accompanying the bloom was a scarcity of zooplankton. However, as levels of R. eriensis peaked in 1985, species of zooplankton and phytoplankton gradually began to reappear so that diversity and species number are approaching historical levels.

Key Words - heavy metals, salmonids, metallothionein, phytoplankton, zooplankton, bioaccumulation.

Introduction

Buttle Lake has been subject to long term heavy metal contamination as a result of a copper/lead/zinc mine which began operation in 1966 (Westmin Resources Ltd.) and is located 6 km west of the south end of the lake adjacent to Myra Creek (Fig. 1). Water chemistry analysis by the Waste Management Branch, B.C. Ministry of Environment (Clark, 1980; Clark and Morrison, 1982) initially identified zinc and copper as the principal hazards to aquatic life throughout the watershed. Concentrations continued to rise, peaking in 1981, when surface concentrations in the south basin of Buttle Lake reached as high as 0.370 mg/L zinc, 0.040 mg/L copper, 0.025 mg/L lead and 0.0036 mg/L cadmium.

A number of studies examined Buttle Lake, the mine site, including Myra Creek and downstream waters before the metals problem was identified (literature summary in Clark and Morrison, 1982). However, once a trend to increasing metal levels was recognized, intensive scientific investigations were launched to evaluate the environmental impact of the mining operation and to determine remedial actions.

In 1981, when metal levels were peaking, salmonid liver and muscle tissue showed elevated levels of metals. In addition, elevated levels of salmonid hepatic metallothionein were correlated with zinc concentrations in Buttle Lake (Roch and McCarter, 1984a). Available data indicated that by 1981, changes in the zooplankton community had taken place so that numbers of cladocerans and calanoid copepods had decreased significantly (Nordin et al., 1985). Austin et al. (1985) found that when comparing samples taken in 1966/67 to those taken in 1980/81/82, that species of phytoplankton and periphyton sensitive to heavy metals had dramatically decreased in abundance, while tolerant forms had increased in numbers. Furthermore, species numbers and diversity had decreased throughout the lake system.

Westmin Resources Ltd. sponsored a number of studies to examine the sources of contamination within the mine site and implemented remedial measures. Interim measures taken by the mine in 1981 included the diversion of the alkaline chlorinated effluent to the tailings ponds to enhance

settling of metals. However, intensive sampling showed that most metal contaminants originated from acid mine drainage below the waste rock areas and the open pit and were not treated before entering Myra Creek (B.C. Research, 1981; B.C. Research, 1982; Ferguson and Menling, 1986). Subsequently, a surface and groundwater collection and treatment system was installed in 1983 (phase one in September, 1982). It was estimated that this system reduced loadings by 80% (Norecol Environmental Consultants, 1984). In July, 1984, an on-land tailings disposal method replaced the system of lake disposal. B.C. Research (1980) had estimated that 2 million tonnes of tailings had been discharged directly into the south basin of Buttle Lake, from 1967 to 1980, (B.C. Research, 1980), resulting in elevated suspended sediments and turbidity, particularly at depth.

As a result of the reduced loadings, it was expected that concentrations of metals throughout Buttle Lake would decline. Utilizing the filling box model described by Killworth and Carmack (1979) to model Buttle Lake it was predicted that zinc levels would return to background levels within three years of implementation of the collection and treatment system (Norecol Environmental Consultants, 1984). However, the model also indicated that some isolated higher concentrations of zinc could persist during this period. Of further concern were potential sources of contamination such as logging, land clearing and partial re-routing of Myra Creek associated with site preparation for on-land tailings disposal. In addition, increased mine production, from 850 t/day to 2,700 t/day by October, 1985, followed by further increases to 3,200 t/day in late 1986, increased the effluent loadings.

Initial studies immediately following the implementation of the remedial measures in 1983 found that the phytoplankton community had become a virtual monoculture and that the zooplankton population was very sparse (Deniseger et al., 1986). Thus, rather than indicating improved water quality, the plankton population actually indicated deteriorating water quality. The present study is the first to cover the transition period from high metals/low plankton diversity to low metals/high plankton diversity.

In addition, the predicted three year period for return to background metal levels is covered. This study examines the existence, rate and extent of recovery of the lake ecosystem from 1983 through 1986 by examining the water chemistry, phytoplankton, zooplankton, and fish in Buttle Lake.

Site Description

Buttle Lake lies in a "U" shaped valley in the mountains of Strathcona Park, Vancouver Island, British Columbia at an altitude of 218 m. With a length of 35 km, surface area of 35.3 km^2 and mean depth of 45 m, it is the largest in a chain of lakes in the Campbell River watershed (area 1404 km^2). Buttle Lake drains northward through a short channel into Upper Campbell Lake. The water levels of both lakes are controlled by B.C. Hydro at Strathcona Dam. The maximum drawdown is generally 8.4 m but, the lake can be drawn down as much as 12 m (Clark and Morrison, 1982; Nordin et al., 1985). The Campbell system continues eastward some 20 km through Lower Campbell Lake, John Hart Lake and the Campbell River which flows to sea (Discovery Passage). For further lake morphometry information see Nordin et al., (1985).

Nutrient levels throughout Buttle Lake are characteristic of oligotrophic lakes on Vancouver Island. The region has a maritime climate and considerable precipitation of 150-300 cm per annum with rapid runoff.

Native fish in Buttle Lake are rainbow trout (Salmo gairdneri), cutthroat trout (Salmo clarkii clarkii), Dolly Varden char (Salvelinus malma), stickleback (Gasterosteus aculeatus) and prickly sculpin (Cottus asper). Anadromous salmonids have never had access to the system upstream of a major waterfall near John Hart Dam.

Methods

Chemical and biological samples were obtained at specific stations in Buttle Lake and Upper Campbell Lake (Figure 1).

Water Chemistry

Samples were taken using a non-metallic Van Dorn water bottle and placed in clean, polyethylene bottles for analysis; those bottles intended for metals analyses were acid washed prior to use. Samples for total metals analyses were field acidified (4 mL HNO_3 per litre of sample). Samples for dissolved metals analyses were filtered through a 0.45 micron millipore filter, followed by preservation with 4 mL HNO_3 per litre of sample. Samples for all other parameters were submitted unfiltered and unpreserved. All samples were sent in coolers to the Environmental Laboratory, where analyses were performed by standard procedures (Clark and Shera, 1985) as outlined in Table 1.

Aquatic Biota

Phytoplankton and zooplankton were sampled qualitatively utilizing horizontal surface tows and 25 m vertical tows with 64 micron mesh plankton nets. Discrete quantitative samples were collected using a non-metallic Van Dorn water bottle. All samples were preserved with 5% buffered formalin. Subsamples of net plankton were examined in the lab by microscope utilizing random transects. Abundance was expressed as a modification of that by McIntyre (1968): abundant (a) observed in more than 80% of the fields; common (c) observed in more than 50% of the fields; frequent (f) observed regularly in at least 10% of the fields; scarce (s) encountered more than once but in less than 10% of the fields; rare (r) encountered only once. Subsamples were taken from quantitative samples and settled in Utermohl chambers with the aid of Lugols solution. Using a minimum numbers graph

(Venrick, 1978), it was determined that the enumeration of 20 microscope fields (at 200x) was sufficient to characterize the community.

Salmonids were collected utilizing two gill nets consisting of a gang of three different mesh sizes (2.5 cm, 5.0 cm, 7.5 cm), each approximately three metres deep, 75 metres long with a weighted bottom line and buoyant mainline. An anchor was utilized to sink the nets at the deep end. The nets were deployed at two sites (one net per site), one 100 metres north of Myra Creek and the other opposite Myra Creek on the east side of Buttle Lake (figure 1). After the fish were removed from the nets, the tissue samples were removed and frozen over dry ice. Liver for metallothionein analysis were taken from freshly killed fish and immediately frozen. Fish lengths, weights, sex, and species were also recorded.

Salmonid hepatic metallothionein concentrations were determined by the Environmental Toxicology Laboratory at the University of Victoria (Roch et al., 1982). Livers were homogenized in ice cold 0.9% NaCl and 10mM Tris buffer pH 8.6 with 8 passes of a stainless steel/teflon Potter-Elvehjem homogenizer and made up to 5 ml of 0.9% NaCl. Large livers (>0.5 g) were homogenized in three volumes of buffer and a 0.2 ml aliquot was made up to 5.0 ml. After heat denaturing at 80°C for five minutes, the homogenates were kept on ice and filtered through a 0.45 micron filter. Metallothionein in the filtrate was determined by differential pulse polarography (Roch and McCarter, 1982).

To assess trends over time within Buttle Lake and to compare Buttle Lake to data for other B.C. Lakes, the fish tissue metals data was grouped into three time periods: 1966-73, the initial period of mine development and operation; 1979-81, the period of greatest impact and highest ambient concentrations; 1983-86, the period following implementation of the ground-water collection and treatment system and decreasing ambient metal levels. No fish were collected from Buttle Lake by the Ministry of Environment and Parks either prior to the mine development, nor in the years between the above three time periods. In addition, similar data for fish collected in 1967-86 from uncontaminated lakes in British Columbia were combined and used

as a control. Statistical comparisons were by student's t-test to determine differences significant at $p < 0.05$ except small data sets which were excluded from consideration.

Results

Available data indicate that prior to the implementation of the collection and treatment system in 1983, concentrations in Myra Creek, downstream of the minesite, were as high as 2.31 mg/L zinc (figure 2), 0.170 mg/L copper, 0.0045 mg/L cadmium and 0.004 mg/L lead (the metals of primary concern). Since that time, levels have decreased considerably, particularly since the spring of 1984, as concentrations now generally remain below 0.5 mg/L zinc, 0.04 mg/L copper, 0.0015 mg/L cadmium and 0.005 mg/L lead. Levels tend to be highest during winter months when high rainfall and rapid runoff results in greater quantities of water percolating through the mine site. In spring, concentrations in Myra Creek begin to drop as high elevation snow-melt contributes to higher flows and dilution. During the relatively dry summer months, levels generally remain below 0.100 mg/L zinc, 0.007 mg/L copper, 0.003 mg/L lead and below detection limits of 0.0005 mg/L cadmium.

Levels of nitrate/nitrite and ammonia, in Myra Creek, have consistently been slightly higher downstream of the mine. However, since August, 1984, a trend to substantially increasing levels of these nutrients has been in evidence (Figure 3). Since September, 1985, total dissolved phosphorus has also periodically increased to levels higher than those found previously, particularly during low flows (Figure 3).

Temperature data for Myra Creek and Buttle Lake indicates that during the winter the flow would be across the lake surface or at shallow depth since the inflow water is similar to or slightly cooler than surface lake water. However, as the lake is warmed during spring, the stream remains relatively cool, partly due to substantial snow melt. As a result, the cooler and thus denser stream water would enter the lake as an interflow at a depth of 10-30 m in midsummer (Nordin et al., 1985). Thus, materials carried by Myra Creek may not always be most evident in surface waters.

The distribution of heavy metals throughout Buttle Lake has shown a pattern with both distance from the minesite and depth. Concentrations of

zinc, copper, cadmium and lead are highest near the mine site and decrease with distance (Figure 4, Table 2). Depth distribution patterns have indicated that metal concentrations are higher in the deeper portions of the lake, particularly in the south basin. For example, in 1980, zinc concentrations peaked at 0.370 mg/L at the surface, and 0.440 mg/L at 30 m. In addition, metals have followed seasonal patterns with highest concentrations occurring during the winter months.

Nordin et al (1985) recommended the following objectives for combinations of zinc, copper and cadmium found in the ratios present in Buttle Lake (400:20:1): 0.05 mg/L zinc, 0.0025 mg/L copper and 0.0013 mg/L cadmium as well as 0.005 mg/L lead. The objective of 0.05 mg/L zinc has been met in surface waters from April 1985 through October 1986. However, despite considerable improvement, zinc levels at depth are frequently higher than the objective. Copper levels continue to exceed the objective, particularly at depth, while lead levels are consistently below the objective. While cadmium levels are generally below the detection limit of 0.0005 mg/L, concentrations may still be higher than the objectives.

The Environment Canada water quality guidleines for the individual metals are 0.03 mg/L zinc, 0.002 mg/L copper, 0.0002 mg/L cadmium, and 0.001 mg/L lead (Environment Canada, 1987). These guidelines are only occasionally met in Buttle Lake, particularly at depth, during the winter.

Obviously, the metal concentrations in Buttle Lake have decreased considerably. However, the mine remains a significant source of heavy metal input as indicated by the levels in Myra Creek (Figure 2). As a result, concentrations in the lake have not reached background levels within the time frame predicted by Norecol (1984). In fact, it is unlikely, given the present metal loadings to Buttle Lake, that background concentrations can be reached.

While heavy metal concentrations have decreased, nutrients in Buttle Lake during 1984 no longer followed patterns typical of oligotrophic lakes of the area. Historical sampling has indicated that nutrients peak during

the winter months and are at their lowest through the summer. In 1984, total dissolved phosphorus reversed its seasonal pattern so that maxima of 0.007 to 0.008 mg/L were maintained through the summer before dropping off in October. Phosphorus levels followed earlier patterns through 1985/86. Occasional peaks of total Kjeldahl nitrogen occurred through 1984, 1985 and 1986 (figure 5, table 3). Prior to May, 1984, silica concentrations showed very little seasonality, generally ranging from 2.5 to 3.0 mg/L throughout the year. However, during the summers of 1984 and 1985 (June through October), concentrations fell below 2.0 mg/L and were as low as 1.6 mg/L, particularly in the south basin. In 1986, concentrations once again ranged 2.5 to 3.0 mg/L.

It appears that the average N:P ratio (total nitrogen: total phosphorus) has increased throughout Buttle Lake since 1983. The N:P ratio in the south basin had increased from 12:1 to 19:1 from 1983 to 1984, while the north end of the lake actually decreased slightly from 18:1 to 16:1. However, by 1985 and through 1986, the entire lake had increased to a range of 22:1 to 27:1.

The cessation of tailings discharge into the south basin of Buttle Lake in 1984 has focused attention on the post depositional behaviour of the tailings deposit. While it has been shown that the tailings are not releasing heavy metals to the overlying lake water (Pedersen, 1983a), the tailings discharge may have affected nutrient cycling within Buttle Lake (Pedersen, 1984). The disposal of lime bearing tailings such as the Westmin discharge (pH 10.5) may limit or reduce productivity by consuming phosphorus and inhibiting phosphorus regeneration. If this is the case in Buttle Lake, the switch to land tailings disposal may benefit lake productivity. Once the tailings deposit becomes covered with an anoxic veener of sediment, the natural phosphorus cycle would resume. However, it has been estimated that it would take 7 to 15 years for a sufficient layer to form (Pedersen, 1983b). Available data through 1986 does not indicate increases in phosphorus either at depth or at the surface.

Cessation of tailings disposal to the lake has removed a major source of turbidity and/or suspended sediments from Buttle Lake. While extinction depth estimates, as measured by secchi depth, have, until recently, shown little evidence of increasing through the summer months, winter estimates appear to have increased, indicating improved water clarity. The phytoplankton bloom throughout the lake likely obscured improvements in extinction depth, particularly during the summer months.

Hepatic metallothionein concentrations in rainbow trout collected in the south basin of Buttle Lake are summarized in Table 4. Levels have decreased from a high of 269 ($\pm$ 23) nanograms/g wet liver in August, 1981 to a low of 64 ($\pm$ 22) nanograms/g wet liver June, 1985 which is comparable to control lakes.

In 1979-81, copper, zinc and lead muscle tissue levels in rainbow trout were significantly higher than the control (Tables 5 & 6). However, by 1983-86, copper and zinc levels had improved to levels similar to the control data set. Available lead data were insufficient for comparison. While cutthroat trout and Dolly Varden muscle tissue data is essentially limited to 1979-81 for Buttle Lake, trends generally follow those for rainbow trout. However, cutthroat zinc levels and Dolly Varden copper levels were not significantly different from the control in 1979-81. In addition, mercury levels in Dolly Varden muscle in 1979-81 were actually significantly lower in Buttle Lake than the control.

When comparing muscle tissue to liver tissue (Table 7), copper, zinc, and cadmium were significantly higher in liver tissue than in muscle tissue for Buttle Lake and the control in all three species. Copper, zinc and cadmium were significantly higher in liver tissue in Buttle Lake and the control in all three species. However, lead and mercury did not always follow this pattern. Lead levels in the control indicated that Dolly Varden liver levels were not significantly different from muscle tissue levels. Lead levels in Buttle Lake fish were significantly higher in liver tissue but this trend was only of borderline significance ($0.05 < p \leq 0.10$) in the control. Available comparative mercury data was limited to the control,

where both Dolly Varden and cutthroat trout liver tissue levels were not significantly different from muscle tissue levels.

Liver tissue metal levels for rainbow trout indicate that copper, zinc and lead levels were significantly higher than the control in 1966-73. Data for 1979-81 was limited to cadmium which was found significantly higher than in the control. By 1983-86, copper and cadmium remained higher than the control while only zinc had improved to levels similar to the control. While Dolly Varden and cutthroat trout data is limited to 1983-86, it follows the rainbow trout data, with the exception of copper in Dolly Varden which was not significantly different from the control although borderline ($0.05 < p \leq 0.10$).

Interspecies comparisons of muscle and liver tissue metal levels (Table 8) show that all three species do not necessarily exhibit similar concentrations. In fact, only zinc and lead levels were similar in all cases (muscle and liver) when comparing rainbow, cutthroat and Dolly Varden. Rainbow trout liver tissue copper levels were significantly higher than both Dolly Varden and cutthroat trout levels in both the control and Buttle Lake. Similarly, cadmium levels were higher in rainbow trout liver in Buttle Lake (insufficient control data). Mercury and copper muscle tissue interspecies comparisons were limited to control data, where Dolly Varden muscle tissue levels were higher than the other two species.

In 1981, the zooplankton community indicated a trend to decreasing numbers of cladocerans and calanoid copepods as the mine was approached. Throughout 1984, zooplankton were present in such low numbers that zooplankton were found in only one sample (July). Zooplankton began to reappear in 1985, gradually becoming more abundant and diverse through the spring and summer months. By late summer, the community was represented by Keratella cochlearis, Daphnia sp., Diaptomus sp., Holopedium gibberellum, Leptodora kindtii, Kellicotia longispina, Bosmina longirostris, Cyclops sp., Polyarthra sp., and Bosmina coregoni. In 1986, additional species such as Polyphemus pediculus and Sida sp. became more abundant.

In 1980/81 the phytoplankton community was atypical of the region's oligotrophic lakes and was dominated by Navicula cryptocephala, Synedra acus, Synedra filiformis, Cyclotella bodanica, and Cyclotella glomerata (Austin et al., 1985; Austin and Munteanu, 1984). As metal levels began to decrease, S. acus and S. filiformis initially became more dominant. In October, 1983 Rhizosolenia eriensis began to appear in substantial numbers, gradually becoming more and more numerous throughout 1984 and well into 1985, comprising a virtual monoculture at times (Deniseger et al., 1986).

Whereas, prior to 1983, cell numbers generally remained below 100 cells per mL, from May, 1983 through June, 1985 cell numbers were consistently higher, peaking at 26,000 cells/mL in June, 1985. Levels gradually decreased through the remainder of 1985 and 1986. By December, 1985 levels were less than 600 cells/mL and in 1986, generally remained below 200 cells/mL. Although R. eriensis remained dominant through this period, other species such as Dinobryon sertularia, became gradually more abundant. Overall species number increased as well, with numerous Chrysophytes, Chlorophytes, Pyrrhophytes and Cyanophytes found.

Discussion

It appears that earlier trends to increasing levels of heavy metals have been reversed and that the water quality of the lake is improving. However, recovery of the biota of Buttle Lake appears more complex than a simple reversal of trends. Furthermore, it appears that fish (i.e. rainbow trout) may be recovering first, as illustrated by metallothionein levels, and are thus less sensitive than zooplankton and phytoplankton.

Wherease in 1979-81 both copper and zinc levels in Buttle Lake rainbow trout muscle tissue were significantly higher than for rainbow collected from non-contaminated lakes in British Columbia, rainbow collected during 1983-86 from Buttle Lake showed improved levels for copper and zinc such that there was no significant difference from non-contaminated lakes. However, copper and cadmium levels in rainbow liver remain statistically greater in 1983-86 than for livers from similar fish collected from non-contaminated lakes.

Trends in the tissue metals data for Dolly Varden char and cutthroat trout are difficult to determine due to the paucity of data. However, it is apparent that trends in rainbow trout data should not be generalized to include all three species. Particular attention should be paid to copper and cadmium, which were accumulated in significantly higher levels in rainbow trout than in either Dolly Varden or cutthroat trout in Buttle Lake.

Metallothionein concentrations showed a better correlation with ambient metal concentrations than do tissue metal concentrations. Roch and McCarter (1984a) report that salmonid hepatic metallothionein concentrations appear to be indicative of the degree of contamination when salmonids are exposed to mixtures of copper, zinc and cadmium approximating ratios found in Buttle Lake. In fact, since metallothionein provides a measure of only the biologically active components of the contaminating metals, it may offer more reliable toxicity estimates than standard bioassays or actual measurement of metal concentrations.

Roch and McCarter (1984b) proposed a metallothionein concentration of 100 nmol per g liver as a safe level for salmonids. In the ratios found in Buttle Lake, this correlates to concentrations of 0.050 mg/L zinc, 0.0025 mg/L copper and less than 0.0005 mg/L cadmium, virtually identical to surface concentrations in June, 1984 when metallothionein levels decreased to 105 (± 50) nmol/g. As metal levels continued to decrease through 1985, metallothionein levels further decreased to 64 (± 22) nmol/g in June, 1985. While metallothionein levels were not measured in 1986, it is likely that continuing decreases in metal concentrations led to corresponding improvements in metallothionein levels. Overall, by 1985, hepatic metallothionein levels had dropped by 76% since 1981 (table 4), as a result of considerably lower metal concentrations. Thus, based on the proposed safe level for salmonids, hepatic metallothionein concentration was well within limits in 1985, indicative of improving water quality and minimal biological stress. In 1981, hepatic metallothionein was measured in rainbow trout captured from Upper Quinsam Lake, a nearby lake with background metal levels, used as a control site. Metallothionein levels were measured as 58 (± 14) nmol/g, not statistically different ($P < 0.05$) from that measured in Buttle Lake in 1985.

Although the fish have been exposed to far more zinc than copper, the metallothionein is composed mainly of copper-thionein (Roch and McCarter, 1984b). They suggested that the ratio of metals in the liver tissue reflects the binding capacity of the metals to thionein (copper > cadmium > zinc). This phenonmenon may explain why zinc levels have decreased in fish livers, whereas copper and cadmium levels remain significantly above fish levels for fish collected from non-contaminated lakes.

The dominant phytoplankton association of Rhizosolenia/Asterionella/Tabellaria/Ceratium/Peridinium in 1966-67 had changed significantly by 1980. Of particular interest was the virtual disappearance of Rhizosolenia eriensis from south Buttle. R. eriensis remained periodically numerous at North Buttle and was abundant (up to 1,228 cells/mL) in John Hart and Lower

Campbell Lakes, the two most downstream lakes in the watershed. It is, thus, likely that the distribution of R. eriensis was limited by the gradient of metals present. However, no data exist in the literature on the sensitivity of R. eriensis to heavy metals.

As metal levels throughout Buttle Lake began to decrease, R. eriensis did not immediately return to prominence. Not until October, 1983, did R. eriensis begin to become abundant. However, at that point, it became so abundant (peaking at 26,000 cells/mL) that for more than one year, from mid 1984 to mid 1985, the surface of Buttle Lake appeared olive brown in colour and extinction depth measured by Secchi disc was depressed during the entire period (figure 6).

Recent lake fertilization studies in coastal oligotrophic British Columbia lakes found that Rhizosolenia spp. respond to changes in N:P ratio as a result of nutrient addition (Hardy et al., 1986). In Hobiton Lake, a shift to large microplankton dominated by Rhizosolenia spp. occurred when the average N:P ratio was 129:1, as compared to 38:1 to 52:1 the previous three years. In Nimpkish Lake, Rhizosolenia spp. became dominant when fertilizer was added in an N:P ratio of 15:1 and then decreased the following year when the N:P ratio was lowered to 1:1. Increases in abundance of Rhizosolenia spp. also occurred following increased dissolved nitrogen loads to Mohun Lake (Perrin et al., 1984). In addition, growth rates of Rhizosolenia spp. are reported to be higher in laboratory cultures with a high N:P ratio (Hardy et al., 1986). An additional contributing factor may be the high chlorophyll: cellular phosphorus ratio in R. eriensis, (Stockner and Shortreed, 1985), indicative of larger increases in biomass accompanying relatively smaller increases in phosphorus.

Available data indicates that N:P ratios initially increased in the south basin of Buttle Lake in 1984 and throughout the lake by 1985 indicative of a source within the south basin and the flushing rate of 1.03 years for Buttle Lake. The source of nitrogen appears to be Myra Creek (Figure 3). Waste water from the mine, as well as clarified sewage treatment plant effluent, is fed through a series of treatment ponds prior

to release to Myra Creek. However, increased explosives residue, sewage and land disturbance arising from increased mine production have likely led to increased nitrogen loading to Myra Creek. In addition, the elevated phosphorus levels found during the summer and fall of 1984 may be related to considerable land disturbance associated with construction of a small dam for power generation, located approximately 5 km southwest of Buttle Lake.

Similar sudden increases in nutrients in a marine oligotrophic environment (Guillard and Kilham, 1977) resulted in prolonged blooms of Rhizosolenia alata. The data also suggest that R. alata may only be capable of such blooms when other faster growing diatoms are inhibited in some way (Guillard and Kilham, 1977). According to Stockner and Shortreed (1985), Pacific coastal lakes are subject to intense grazing on zooplankton by fish, reducing zooplankton abundance significantly so that zooplankton have a relatively minor effect on controlling algal biomass. Consequently, increasing phosphorus loads are accompanied by rapid increases in phytoplankton biomass.

The reappearance of R. eriensis throughout Buttle Lake is likely related to the long term presence of heavy metals. As metal levels decreased throughout the watershed, very tolerant forms lost their competitive advantage allowing other more sensitive forms to repopulate. Lower species number and thus less competition likely contributed to the initial increases in abundance of R. eriensis and may have exaggerated the effects of higher N:P ratios. For example, Rhizosolenia spp. were not dominant in the lake fertilization studies at average N:P ratios ranging from 38:1 to 52:1, higher than those found in Buttle Lake in 1985-86 (22:1 to 27:1). However, it should be noted that occasionally higher ratios, up to 57:1, were found in Buttle Lake.

The nature and abundance of grazers may also have affected the bloom of R. eriensis. For example, a system such as Buttle Lake, which has been dominated by microzooplankton, favours larger diatoms such as R. eriensis (Guillard and Kilham, 1977). Furthermore, freshwater grazers show a

substantial preference for particles less than 30 microns in length and width (Gliwicz, 1977; Porter, 1977). Thus, the lack of large herbivorous zooplankton combined with the large size of R. eriensis to further increase the bloom.

It appears that the bloom of R. eriensis may have resulted from a set of environmental variables occurring during a transition period in water quality associated with decreasing heavy metals, nutrient increases and the long term effects of heavy metal contamination. The conditions appear to have been ideal for the growth of R. eriensis. R. eriensis appears to be tolerant of combinations of metals at ratios found in Buttle Lake approximately up to 0.080 mg/L zinc, 0.010 mg/L copper, less than 0.005 mg/L cadmium. These concentrations appear low enough to permit the growth of R. eriensis yet inhibit many sensitive species which were previously found in Buttle Lake. As metal levels continue to decrease, R. eriensis appears to be gradually losing its competitive advantage. Since the collapse of the bloom in mid summer 1985, total numbers have generally decreased to less than 200 cells/mL, or less than 1% of the peak in June 1985. As a result, water clarity has returned, particularly during the summer months when Secchi depths of up to 18 metres were recorded in 1986. As R. eriensis has declined in abundance and dominance, other species such as Dinobryon spp. have become seasonal co-dominants (spring), while sensitive species such as Tabellaria fenestrata, Tabellaria flocculosa, Fragilaria crotonensis and Asterionella formosa have begun to return. A secondary benefit that may have resulted from the bloom of R. eriensis is the bioaccumulation of metals. Since a large proportion of these cells will eventually settle out of the water column, the bloom may have contributed to decreasing heavy metal levels.

Paralleling the decline of R. eriensis and the continuing metals decrease was the gradual recovery of the zooplankton community throughout Buttle Lake. By late summer 1985, species such as Leptodora kindtii, Holopedium gibberellum, Polyphemus pediculus and Daphnia spp., which had been absent for a number of years, were reappearing. In fact, by 1986,

species composition and diversity were similar to that sampled in 1966-67 (Whately, 1969). The improved zooplankton species composition and abundance in Buttle Lake is likely of benefit to the fisheries. Nilsson and Northcote (1981) found that Daphnia was a major dietary component of rainbow trout at certain times of the year in Upper Campbell Lake. Large zooplankton species such as Daphnia are likely of considerable importance to fish production in lakes such as Buttle Lake, which have limited littoral benthic production as a result of morphometry and large fluctuations in water level.

Thus, the data presented here indicates that water quality in Buttle Lake appears to be improving. However, the fragile nature of the recovery process is illustrated by the prolonged bloom of R. eriensis and the combination of factors which likely led to it. Hepatic metallothionein levels indicate that biological stress upon fish may have been reduced to near background levels while the zooplankton community is beginning to approximate historical species composition. However, copper and cadmium liver tissue levels remain relatively high. While metal levels in the lake have substantially decreased, they have not yet reached background levels as predicted by the filling box model (Norecol, 1984). It is expected that concentrations will continue to decline as the lake reaches equilibrium. It appears that the phytoplankton community, due to its sensitive nature, will require continued decreases in heavy metal concentrations before previously dominant species such as Asterionella formosa, Tabellaria fenestrata and Tabellaria flocculosa can return to prominence. It is also apparent that biological recovery processes lag behind improvements in water chemistry.

Acknowledgements

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Table 1. Methods of water quality analyses employed*

Parameter	Test method employed for majority of analyses
Ammonia	Colorimetric: orthotolidine
Extinction Depth	Secchi disc
Hardness	EDTA titration
NO_2/NO_3	Colorimetric: cadmium reduction plus diazotization.
Orthophosphate and silica (reactive)	Colorimetric: ascorbic acid reduction
pH	Glass electrode
Total phosphorus	Colorimetric: digestion plus ascorbic acid reduction
Cadmimum, copper, lead, and zinc	Dissolved: atomic absorption Total: prior to late 1979 HNO_3/HCl digestion followed by APDC extraction and atomic absorption; since late 1979 HNO_3/HCl digestion followed by atomic absorption (or plasma emission spectrometry for zinc).

*see Clark and Shera (1985) for detailed methodology.

Table 2. Summarized concentrations of heavy metals in surface waters of Buttle Lake. Range of values throughout year with upper values generally found during winter months. Total number of samples in brackets. All values in micrograms/L.

	Station 0130082	Station 0130090	Station 0130088
Total Cadmium			
1980	0.5* - 13 (6)	0.5* - 1.2 (6)	0.5* - 1.3 (3)
1981	0.6 (1)	0.5* - 0.5 (5)	0.6 (1)
1982	0.5* - 0.6 (2)	0.5* - 0.5 (4)	0.5* (1)
1983	0.5* (5)	0.5* (5)	0.5* (5)
1984	0.5* - 0.5 (7)	0.5* (7)	0.5* - 0.5 (7)
1985	0.5* (8)	0.5* - 0.6 (8)	0.5* - 1.5 (7)
1986	0.5* (9)	0.5* (8)	0.5* - (8)
Total Copper			
1980	5 - 30 (6)	4 - 25 (6)	5 - 10 (8)
1981	4 - 30 (5)	7 - 20 (5)	2 - 7 (7)
1982	4 - 15 (5)	4 - 12 (5)	4 - 7 (4)
1983	3 - 9 (5)	3 - 7 (5)	2 - 9 (5)
1984	2 - 20 (8)	3 - 5 (8)	3 - 4 (8)
1985	1* - 7 (8)	1* - 4 (8)	1* - 6 (7)
1986	1 - 10* (9)	1 - 10 (8)	2 - 10 (8)
Total Zinc			
1980	110 - 370 (6)	110 - 290 (6)	85 - 143 (8)
1981	100 - 280 (5)	115 - 220 (5)	40 - 130 (7)
1982	30 - 170 (3)	30 - 125 (3)	100 (1)
1983	18 - 120 (5)	19 - 100 (5)	19 - 100 (5)
1984	10 - 140 (8)	40 - 80 (8)	40 - 80 (8)
1985	30 - 70 (8)	30 - 60 (8)	24 - 50 (7)
1986	20 - 50 (9)	20 - 50 (8)	12 - 40 (8)
Total Lead			
1980	1* - 4 (6)	1* - 2 (7)	1* - 1 (6)
1981	1* - 4 (1)	1* - 4 (4)	1* - 3 (2)
1982	1* - 4 (2)	1* - 3 (3)	1* (2)
1983	1* - 3 (6)	1* - 4 (7)	1* - 3 (6)
1984	1* - 2 (6)	1* - 3 (7)	1* - 2 (6)
1985	1* - 1 (5)	1* - 4 (6)	1* - 4 (7)
1986	1* - 4 (9)	1* - 4 (9)	1* - 2 (8)

* denotes below detection limits

Table 3. Summarized concentrations of nutrients in Buttle lake surface waters. Range of values found throughout the year with upper values generally during winter months. Total number of samples in brackets. All values in mg/L.

Year	Nitrite/ Nitrate	Total Kjeldahl Nitrogen	Total Dissolved Phosphorus	Silica
1981	0.02* - 0.05 (37)	0.01* - 0.11 (21)	0.003* - 0.007 (21)	2.0 - 3.8 (27)
1982	0.04 - 0.06 (10)	0.01* - 0.13 (3)	0.003* - 0.003 (3)	3.0 - 3.7 (6)
1983	0.02* - 0.16 (26)	0.01* - 0.14 (24)	0.003* - 0.005 (25)	1.1 - 3.2 (25)
1984	0.02* - 0.13 (37)	0.01* - 0.19 (35)	0.003* - 0.008 (37)	1.6 - 3.2 (37)
1985	0.02* - 0.09 (41)	0.01* - 0.23 (41)	0.003* - 0.005 (41)	1.7 - 5.7 (40)
1986	0.02* - 0.07 (32)	0.03 - 0.15 (32)	0.003* - 0.004 (32)	2.5 - 3.0 (32)

*denotes below detection limits.

Table 4. Concentrations of metallothionein in rainbow trout captured in the south basin of Buttle Lake.

Date	Metallothionein (nanomoles/g wet liver)
July, 1981	213 ± 61 (n = 6)
August, 1981	269 ± 23 (n = 5)
June, 1983	205 ± 23 (n = 2)
June, 1984	105 ± 50 (n = 17)
June, 1985	64 ± 22 (n = 12)

Table 5. Fish Tissue Metal Content - Summary Statistics (micrograms/g dry wt) (from Clark, 1987)

		RAINBOW TROUT			CUTTHROAT TROUT			DOLLY VARDEN		
a) zinc		Mean	St.dev.	No.	Mean	St.dev.	No.	Mean	St.dev.	No.
Muscle Tissue										
B.C. Lakes	1967-86	18.3	16.4	140	17.9	15.8	81	18.7	16.8	82
Buttle Lake	1966-73	18.3	16.5	26	----	----	--	----	----	--
	1979-81	37.8	34.0	49	24.3	21.9	14	39.2	35.3	37
	1983-86	21.7	19.5	18	18.8	16.9	4	35.2	31.4	5
	1966-86	29.2	26.3	93	23.1	20.8	18	38.7	34.8	42
Liver Tissue										
B.C. Lakes	1967-86	108.6	97.7	180	110.6	99.5	114	133.1	119.8	75
Buttle Lake	1966-73	143.7	129.3	37	----	----	--	----	----	--
	1979-81	168.0	151.2	5	117.0	105.3	2	270	----	1
	1983-86	110.9	99.8	22	108.8	97.9	4	162.3	146.1	14
	1966-86	134.3	120.9	64	111.5	100.4	6	169.4	152.5	15
b) copper		Mean	St.dev.	No.	Mean	St.dev.	No.	Mean	St.dev.	No.
Muscle Tissue										
B.C. Lakes	1967-86	1.07	1.84	134	0.97	1.08	87	2.24	2.23	78
Buttle Lake	1966-73	2.84	1.25	26	----	----	--	----	----	--
	1979-81	2.10	1.64	49	2.15	1.02	14	2.05	1.07	37
	1983-86	0.61	1.68	18	1.62	1.62	4	1.30	1.80	5
	1966-86	2.01	1.72	93	2.03	1.19	18	1.96	1.18	42
Liver Tissue										
B.C. Lakes	1967-86	207.4	179.0	191	90.3	119.0	76	80.4	162.4	75
Buttle Lake	1966-73	529.7	453.5	37	----	----	--	----	----	--
	1979-81	539.0	124.0	5	170.0	137.1	2	47.0	----	1
	1983-86	510.4	307.5	22	587.2	369.3	4	172.6	87.1	14
	1966-86	523.8	387.4	64	448.1	363.3	6	164.2	98.0	15
c) cadmium		Mean	St.dev.	No.	Mean	St.dev.	No.*	Mean	St.dev.	No.*
Liver Tissue										
B.C. Lakes	1967-86	1.17	1.16	107	1.01	0.99	75/3	1.04	0.93	49/8
Buttle Lake	1966-73	----	----	----	----	----	--	----	----	--
	1979-81	19.4	6.1	5	7.5	8.22	4/2	13.0	----	1/1
	1983-86	10.1	4.3	22	5.5	3.10	4/4	5.8	3.0	14/14
	1966-86	11.8	5.8	27	6.5	5.84	9/6	6.3	3.4	15/15

Table 5. Fish Tissue Metal Content - Summary Statistics (micrograms/g) continued

		RAINBOW TROUT			CUTTHROAT TROUT			DOLLY VARDEN		
d) mercury (wet wt)		Mean	St.dev.	No.	Mean	St.dev.	No.	Mean	St.dev.	No.
Muscle Tissue										
B.C. Lakes	1967-86	0.08	0.08	120	0.28	0.37	55	0.12	0.14	48
Buttle Lake	1966-73	0.07	0.04	26	----	----	--	----	----	--
	1979-81	0.06	0.04	34	0.12	0.12	6	0.05	0.03	35
	1983-86	----	----	--	----	----	-	----	----	--
	1966-86	0.06	0.04	60	0.12	0.12	6	0.05	0.03	35
Liver Tissue										
B.C. Lakes	1967-86	0.15	0.22	24	0.22	0.36	24	0.12	0.18	8
Buttle Lake	1966-73	0.06	0.10	2	----	----	-	----	----	-
	1979-81	----	----	--	2.00	----	1	----	----	-
	1983-86	----	----	--	----	----	-	----	----	-
	1966-86	0.06	0.10	2	2.00	----	1	----	----	-
e) lead		Mean	St.dev.	No.	Mean	St.dev.	No.	Mean	St.dev.	No.
Muscle Tissue										
B.C. Lakes	1967-86	1.23	1.00	114	1.18	0.96	49	1.20	0.98	51
Buttle Lake	1966-73	----	----	--	----	----	--	----	----	--
	1979-81	1.90	1.89	49	1.93	2.76	14	2.30	1.56	37
	1983-86	----	----	--	----	----	--	----	----	--
	1966-86	1.90	1.89	49	1.93	2.76	14	2.30	1.56	37
Liver Tissue										
B.C. Lakes	1967-86	1.67	2.29	156	1.64	1.36	84	1.31	1.14	54
Buttle Lake	1966-73	5.47	5.24	11	----	----	--	----	----	--
	1979-81	----	----	--	<1	----	2	<1	----	1
	1983-86	3.14	5.74	7	<1	----	1	3.25	2.06	4
	1966-86	4.56	5.40	18	<1	----	3	2.80	2.05	5

Table 6. Statistical Comparison of Buttle Lake Fish Tissue Metal Concentrations to other B.C. Lakes [x = no significant difference ($P < 0.05$), in = insufficient data, ( ) = borderline significance ($0.05 < P \leq 0.10$) RB = rainbow trout, CT = cutthroat trout, DV = Dolly Varden]

a) copper

	MUSCLE			LIVER		
	RB	CT	DV	RB	CT	DV
1966-73	BUT > BC	--	--	BUT > BC	--	--
1979-81	BUT > BC	BUT > BC	X	in	in	in
1983-86	X	in	in	BUT > BC	in	(BUT > BC)
1966-86	BUT > BC	BUT > BC	X	BUT > BC	in	(BUT > BC)

b) zinc

	MUSCLE			LIVER		
	RB	CT	DV	RB	CT	DV
1966-73	X	--	--	BUT > BC	--	--
1979-81	BUT > BC	X	BUT > BC	in	in	in
1983-86	X	in	in	X	in	X
1966-86	BUT > BC	X	BUT > BC	BUT > BC	in	X

c) cadmium

	MUSCLE			LIVER		
	RB	CT	DV	RB	CT	DV
1966-73	--	--	--	--	--	--
1979-81	--	--	--	BUT > BC	in	in
1983-86	--	--	--	BUT > BC	in	BUT > BC
1966-86	--	--	--	BUT > BC	BUT > BC	BUT > BC

d) mercury

	MUSCLE			LIVER		
	RB	CT	DV	RB	CT	DV
1966-73	X	--	--	in	--	--
1979-81	X	in	BC > BUT	--	in	--
1983-86	--	--	--	--	--	--
1966-86	(BUT > BC)	in	BC > BUT	in	in	--

e) lead

	MUSCLE			LIVER		
	RB	CT	DV	RB	CT	DV
1966-73	--	--	--	BUT > BC	--	--
1979-81	BUT > BC	X	BUT > BC	--	in	in
1983-86	--	--	--	in	in	in
1966-86	BUT > BC	X	BUT > BC	BUT > BC	in	in

Table 7. Comparison of Liver and Muscle Tissue Metal Concentrations [RB = rainbow trout, CT = cutthroat trout, DV = Dolly Varden, x = no significant difference ($p \leq 0.05$), in = insufficient data, parentheses indicates borderline significance ($0.05 < p \leq 0.10$)].

a) copper

	RB	CT	DV
BC Lakes	Liver > Muscle	Liver > Muscle	Liver > Muscle
Buttle	Liver > Muscle	Liver > Muscle	Liver > Muscle

b) zinc

	RB	CT	DV
BC Lakes	Liver > Muscle	Liver > Muscle	Liver > Muscle
Buttle	Liver > Muscle	Liver > Muscle	Liver > Muscle

c) cadmium

	RB	CT	DV
BC Lakes	--	--	--
Buttle	Liver > Muscle	Liver > Muscle	Liver > Muscle

d) mercury

	RB	CT	DV
BC Lakes	Liver > Muscle	X	X
Buttle	in	in	--

e) lead

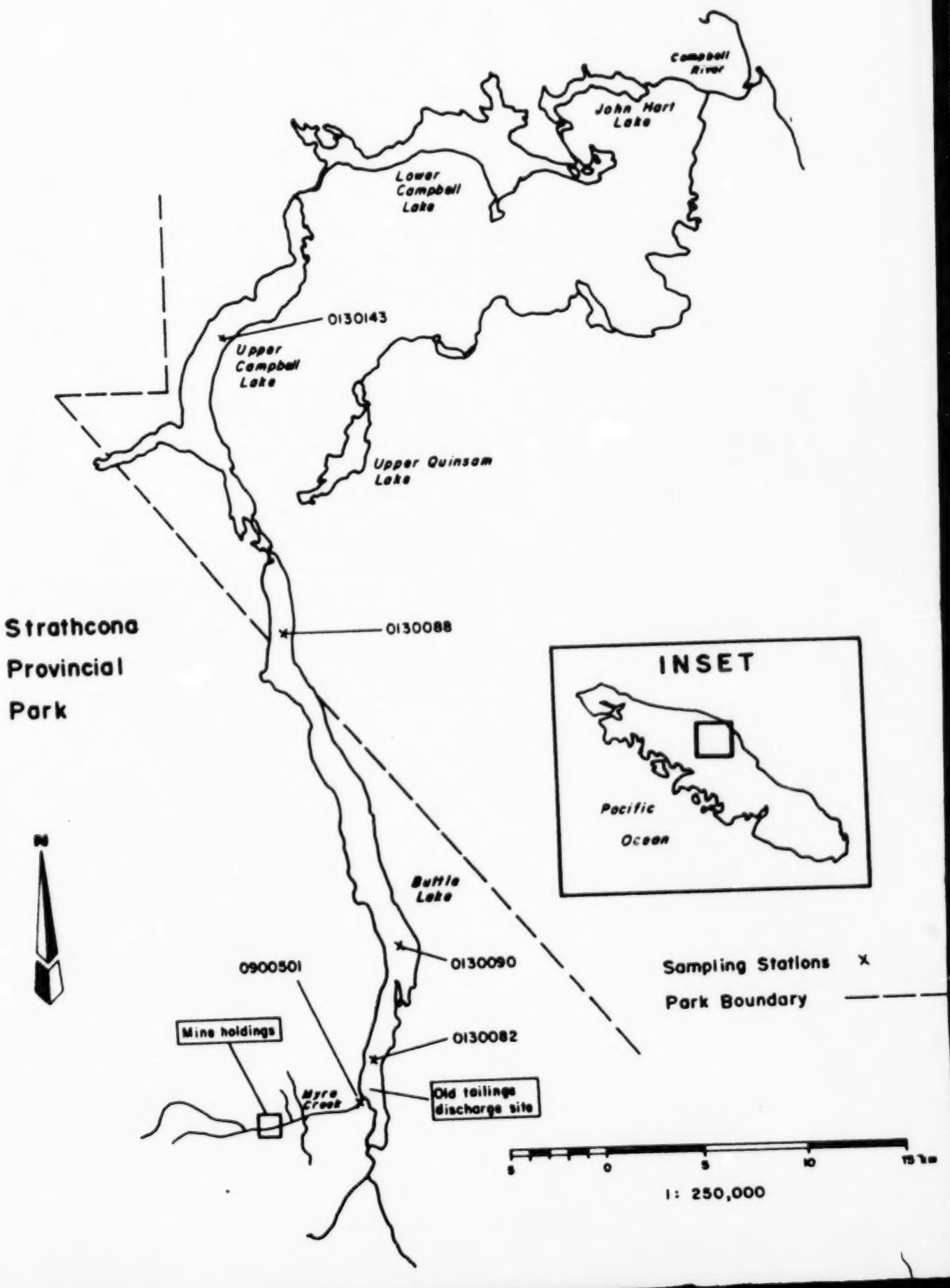
	RB	CT	DV
BC Lakes	(Liver > Muscle)	Liver > Muscle	X
Buttle	Liver > Muscle	in	in

Table 8. Interspecies Comparison of Tissue Metal Levels [RB = rainbow trout, CT = cutthroat trout, DV = Dolly Varden, x = no significant difference ($p < 0.05$), in = insufficient data, parentheses indicates borderline significance ($0.05 < p \leq 0.10$)].

		MUSCLE			LIVER		
a) copper	RB/CT	RB/DV	CT/DV	RB/CT	RB/DV	CT/DV
BC Lakes	X	DV > RB	DV > CT	RB > CT	RB > DV	X
Buttle	X	X	X	in	RB > DV	in
b) zinc						
BC Lakes	X	X	X	X	(DV > RB)	X
Buttle	X	(DV > RB)	(DV > CT)	in	X	in
c) cadmium						
BC Lakes	--	--	--	--	--	--
Buttle	--	--	--	RB > CT	RB > DV	X
d) mercury						
BC Lakes	CT > RB	DV > RB	DV > CT	X	X	X
Buttle	in	X	in	in	--	--
e) lead						
BC Lakes	X	X	X	X	X	X
Buttle	X	X	X	in	in	in

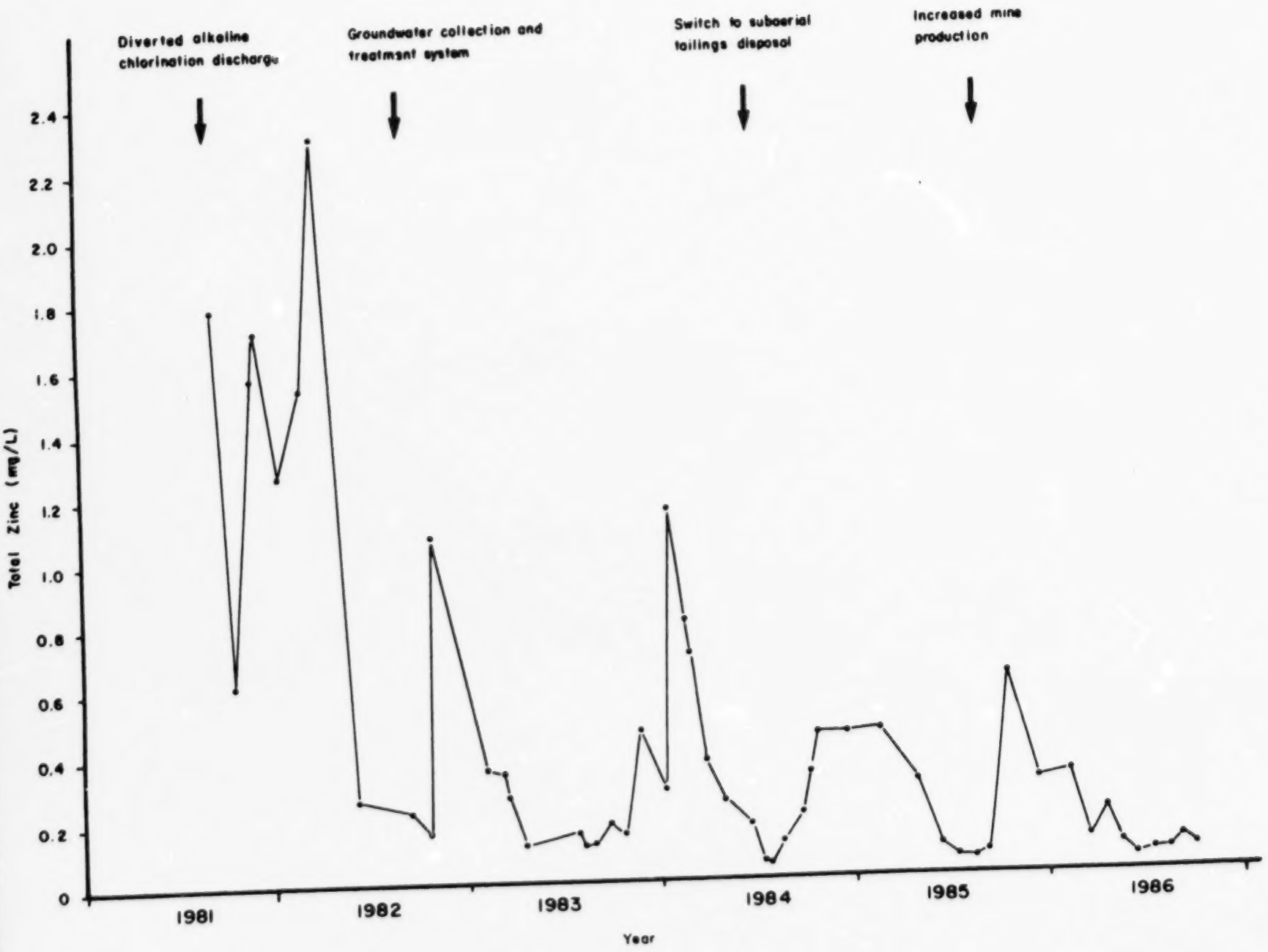
Figures

Figure 1. Map of Buttle Lake showing position of sampling stations and some features of the surrounding watershed.


Campbell River
John Hart Lake
Lower Campbell Lake
0130143
Upper Campbell Lake
Upper Quinsam Lake
Strathcona Provincial Park
0130088
INSET
Pacific Ocean
N
Buttle Lake
0130090
0900501
Sampling Stations
Park Boundary
Mine holdings
0130082
Myra Creek
Old tailings discharge site
0
5
10
1: 250,000

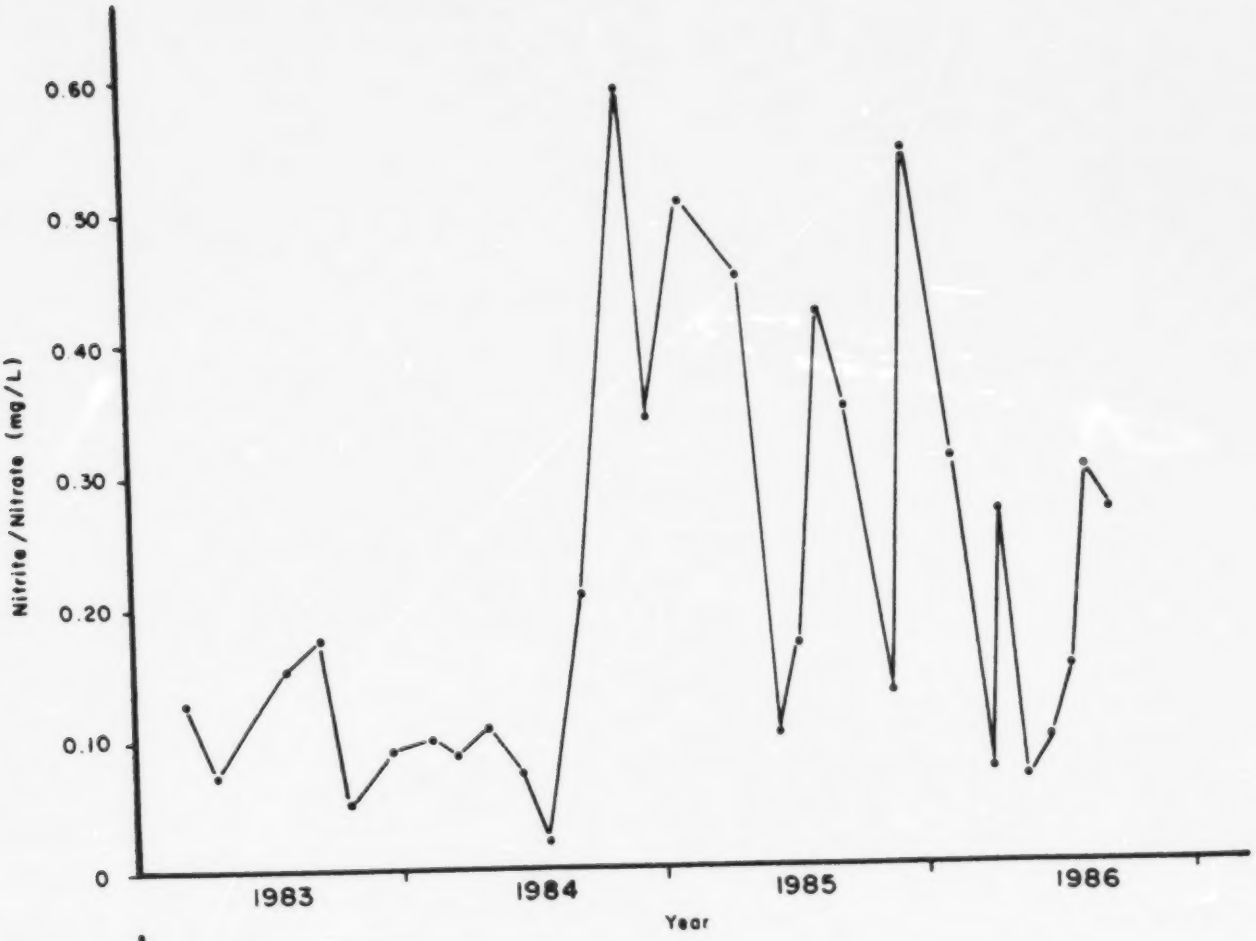
Figures

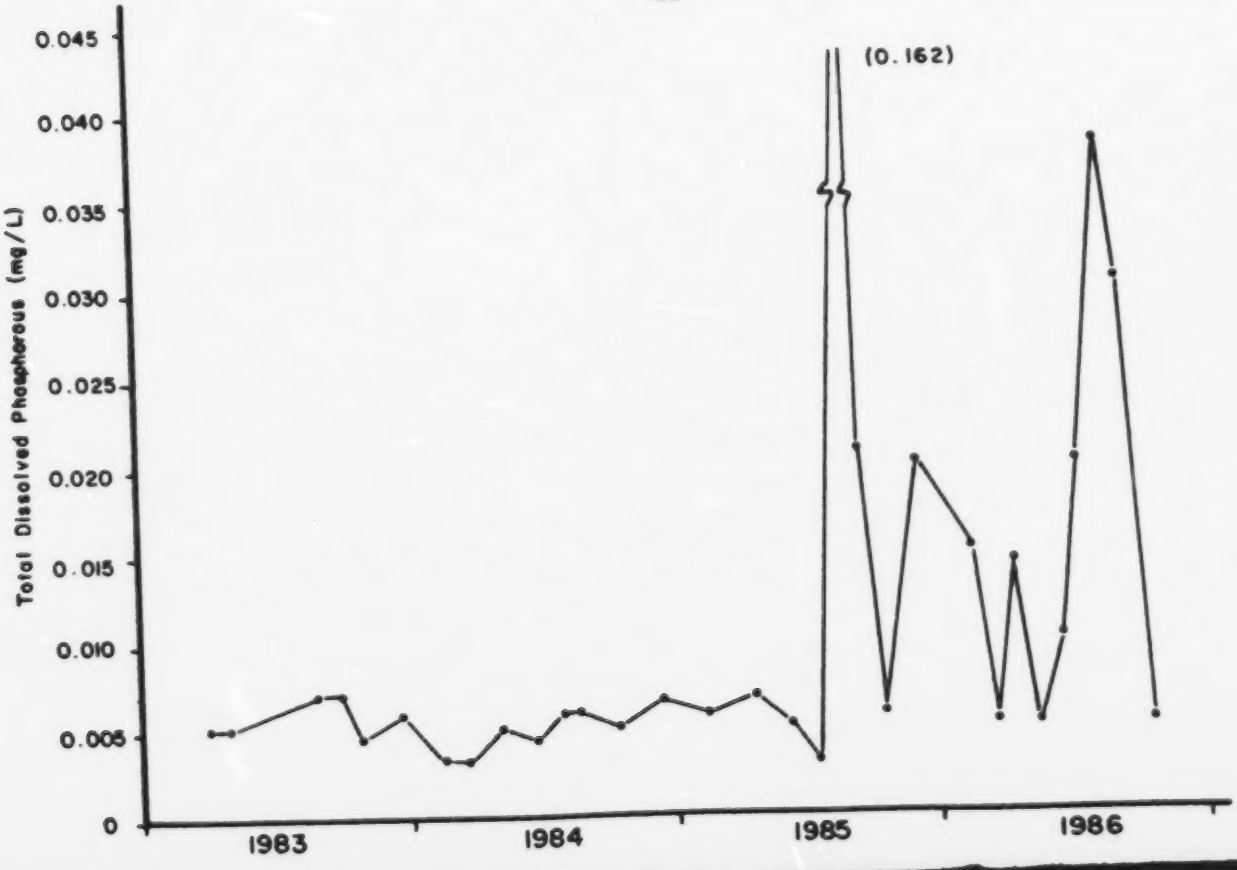
Figure 2. Graph illustrating the range and seasonality of total zinc in Myra Creek at site 0900501.


Diverted alkaline chlorination discharge
Groundwater collection and treatment system
Switch to subaerial tailings disposal
Increased mine production
Total Zinc (mg/L)
2.4
2.2
2.0
1.8
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0
1981
1982
1983
1984
1985
1986
Year

Figures

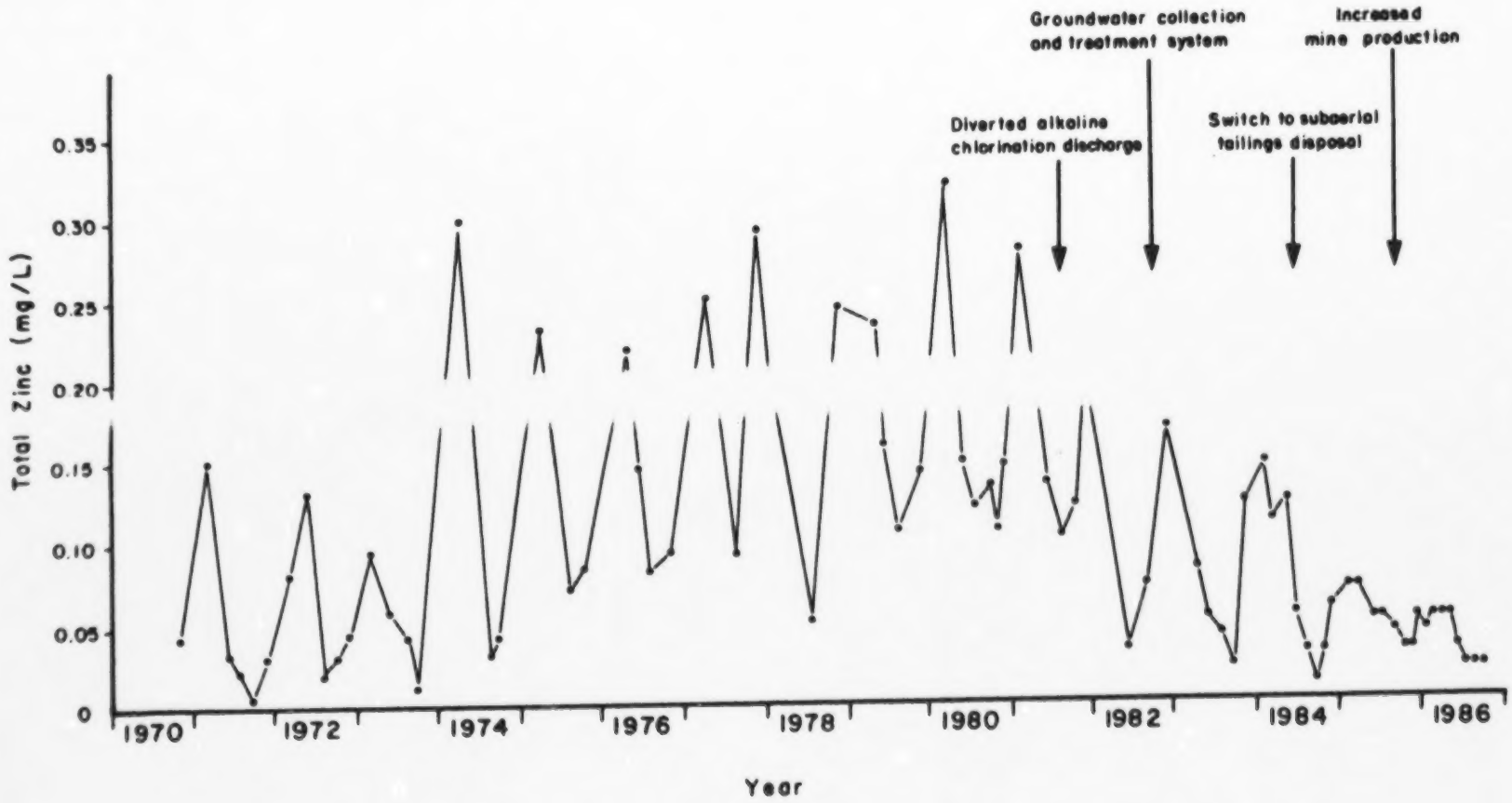
Figure 3. Graphs illustrating the range of nitrite/nitrate and total phosphorus in Myra Creek at site 0900501.


0.60
0.50
0.40
0.30
0.20
0.10
0
Nitrite / Nitrate (mg/L)
1983
1984
1985
1986
Year


0.045
0.040
0.035
0.030
0.025
0.020
0.015
0.010
0.005
0
Total Dissolved Phosphorous (mg/L)
(0.162)
1983
1984
1985
1986

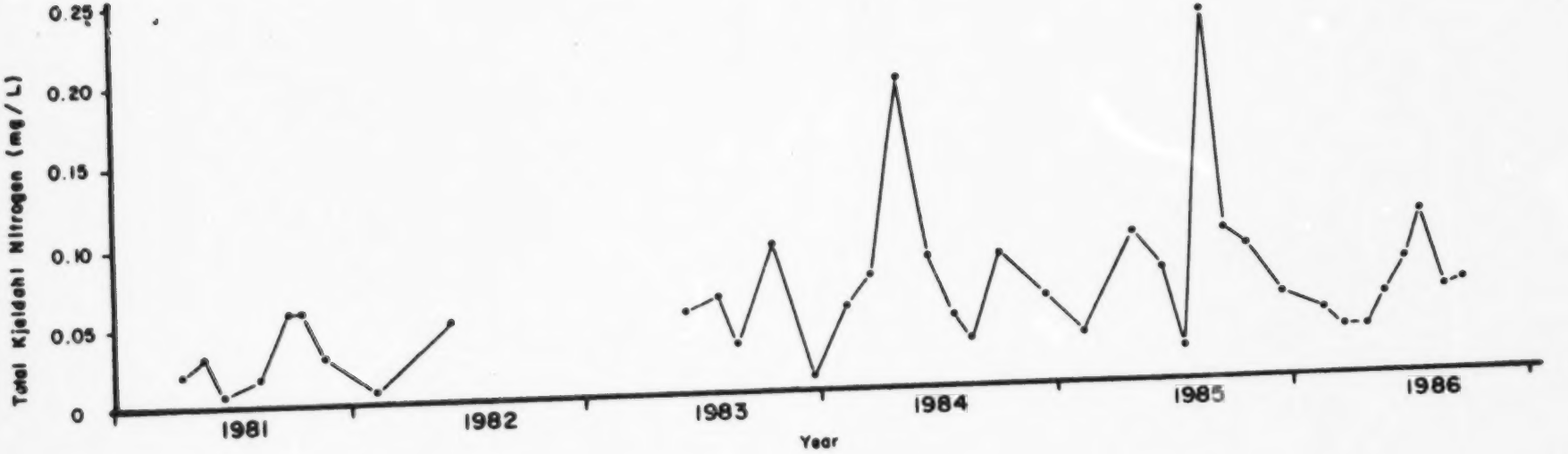
Figures

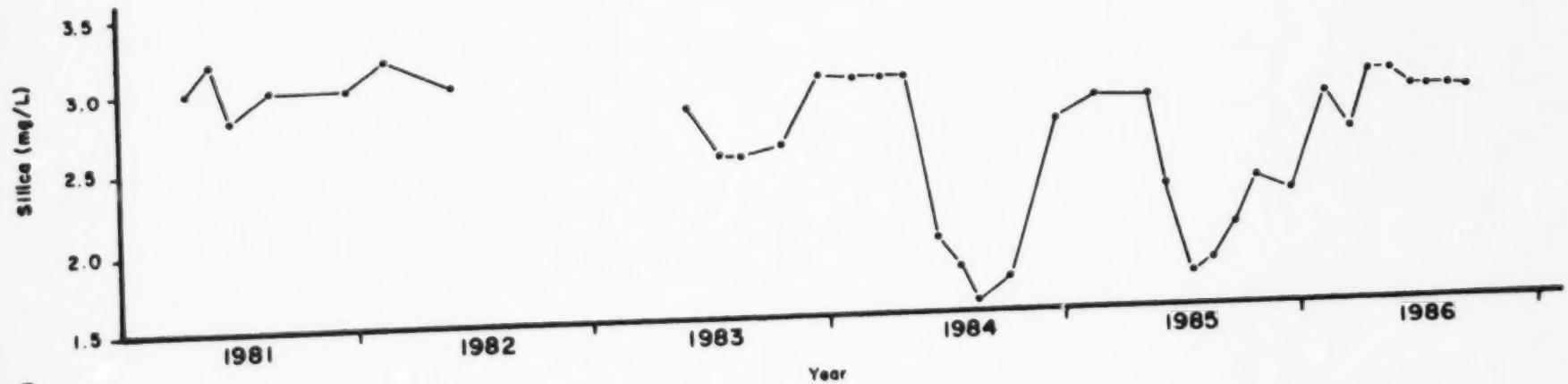
Figure 4. Graph illustrating the range and seasonality of total zinc (surface) at station 0130082, in the south basin of Buttle Lake from 1970 to 1986.

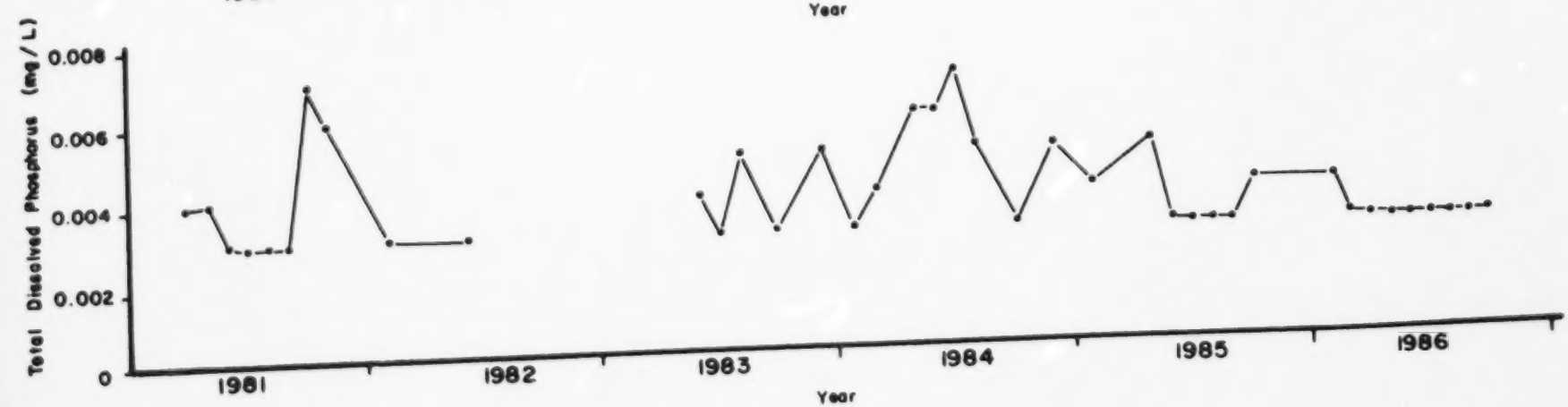

Groundwater collection and treatment system
Increased mine production
Diverted alkaline chlorination discharge
Switch to subaerial tailings disposal
Total Zinc (mg/L)
0.35
0.30
0.25
0.20
0.15
0.10
0.05
0
1970
1972
1974
1976
1978
1980
1982
1984
1986
Year

Figures

Figure 5. Graphs illustrating the range and seasonality of total Kjeldahl nitrogen, silica and total dissolved phosphorus at station 0130082, in the south basin of Buttle Lake from 1981 to 1986.

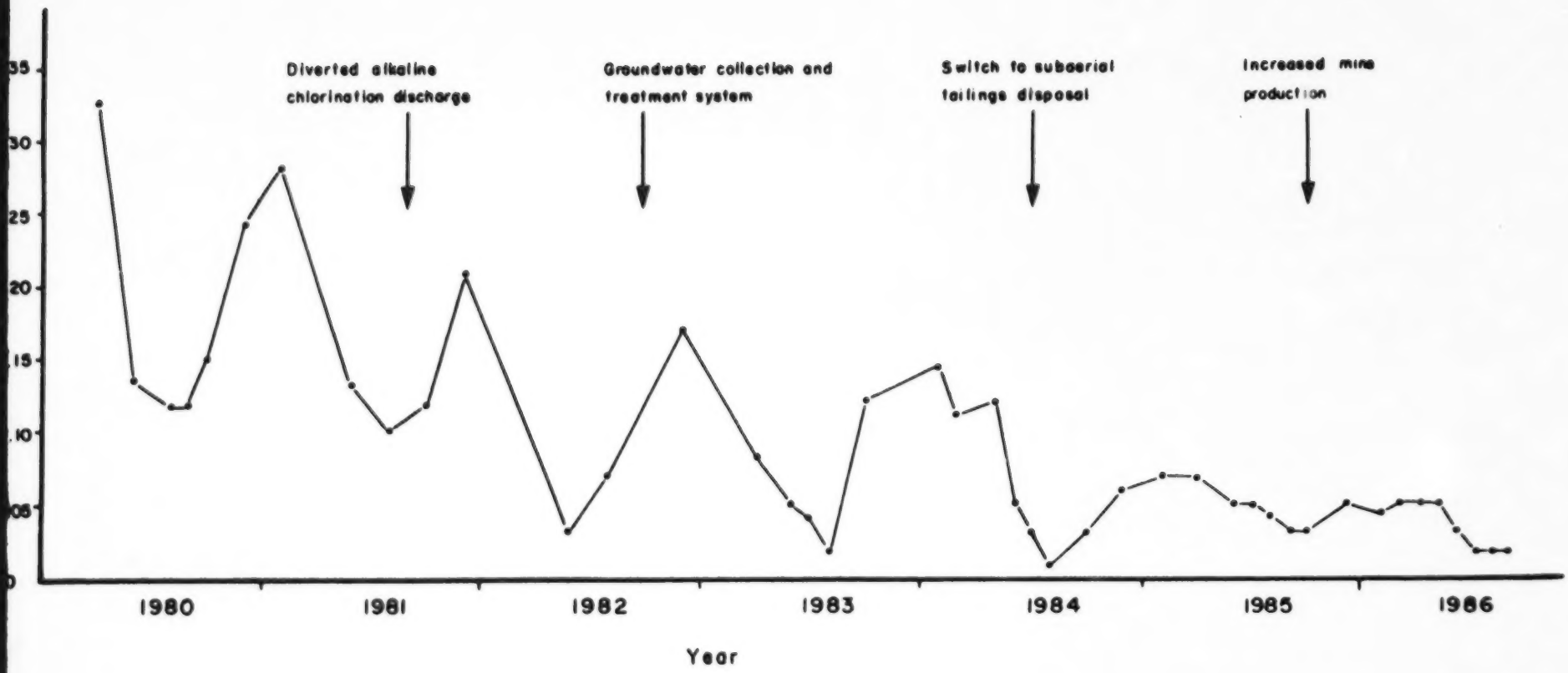

Total Kjeldahl Nitrogen (mg/L)
0.25
0.20
0.15
0.10
0.05
0
1981
1982
1983
1984
1985
1986
Year


Silica (mg/L)
3.5
3.0
2.5
2.0
1.5
1981
1982
1983
1984
1985
1986
Year


Total Dissolved Phosphorus (mg/L)
0.008
0.006
0.004
0.002
0
1981
1982
1983
1984
1985
1986
Year

Figures

Figure 6. Graph illustrating extinction depth (as measured by secchi disc) from 1980 to 1986 at site 0130082, situated in the south basin of Buttle Lake.


Diverted alkaline chlorination discharge
Groundwater collection and treatment system
Switch to subaerial tailings disposal
Increased mine production
35
30
25
20
15
10
05
0
1980
1981
1982
1983
1984
1985
1986
Year